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REVIEW ARTICLE

The capacity of aquatic macrophytes for phytoremediation and their disposal with specific reference to water hyacinth

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Abstract The actual amount of fresh water readily accessible for use is <1 % of the total amount of water on earth, and is expected to shrink further due to the projected growth of the population by a third in 2050. Worse yet are the major issues of water pollution, including mining and industrial waste which account for the bulk of contamination sources. The use of aquatic macrophytes as a cost-effective and eco-friendly tool for phytoremediation is well documented. However, little is known about the fate of those plants after phytoremediation. This paper reviews the options for safe disposal of waste plant biomass after phytoremediation. Among the few mentioned in the literature are briquetting, incineration and biogasification. The economic viability of such processes and the safety of their economic products for domestic use are however, not yet established. Over half of the nations in the world are involved in mining of precious metals, and tailings dams are the widespread legacy of such activities. Thus, the disposal of polluted plant biomass onto mine storage facilities such as tailing dams could be an interim solution. There, the material can act as mulch for the establishment of stabilizing vegetation and suppress dust. Plant decomposition might liberate its contaminants, but in a site where containment is a priority.

Keywords Tailings dam · Macrophytes · Decomposition · Pollution · Phytoremediation · Safe disposal

Introduction

Water is the source of life and has no substitute. Considering that our planet is 70 % covered by water, it is a shock to realize that the actual amount of fresh water readily accessible for human use in the world is <1 % (Postel et al. 1996). Currently a third of the world's population lives under water stress and the future is bleak as the figure is expected to double by 2025 at the current rate of water consumption (Arnell 1999). Furthermore, water scarcity is also likely to increase due to the impacts of climate change (Arnell 2004). It is therefore, imperative that we pay the utmost attention to management and conservation of our renewable surface waters which will remain the main source of domestic and agricultural water supplies, particularly in developing countries. While determining the sources of water pollution and preventing them from reaching our water stores has to be the focal point of any solution, cleaning water by conventional means (chemical, physical and biological) or by phytoremediation is inevitably becoming more important. Conventional methods are however, often less cost-effective and less eco-friendly than phytoremediation where plants are used to remove contaminants from soil or water (Ahluwalia and Goyal 2007). Nevertheless, despite increased public interest in the method, particularly in the last three decades (Henry 2000), its practical use is curtailed by a number of factors, among which is the fate of the phytoremediating plants after use which about little is known. This review investigates the use of aquatic macrophytes in phytoremediation and options for their safe disposal after the process of phytoremediation. Special reference to water hyacinth, *Eichhornia crassipes* (Mart.) Solms-

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Laubach is made because of its wide distribution and reputation as an invasive weed, in contrast to its ability to remove pollutants from water.

Sources of water pollution

While water pollution also occurs through natural physical weathering of geological structures and leaching by runoff, the water pollution that results from a number of anthropogenic activities such as mining, industrial and agricultural practices is unprecedented and remains a major issue of concern (Sood et al. 2012). Disposal of untreated sewage and effluents into surface water is still the norm in many countries (Ismail and Beddri 2009). Globally, an estimated 80 % of used water is neither collected nor treated and is simply discharged into our waterways (Corcoran et al. 2010). The water bodies in the state of Lagos in Nigeria are used as waste water reservoirs by the nearby medium and large scale industries (Anetekhai et al. 2007). Both organic and inorganic contaminants of water from such activities put all aquatic life and human health at risk and particularly threaten developing countries, where between 75 and 90 % of their populations are exposed to unsafe drinking water (Sood et al. 2012). The common contaminants include heavy metals, radionuclides, nitrates, phosphates, inorganic acids and organic chemicals (Arthur et al. 2005). The water pollutants of major concern are the heavy metals such as lead, arsenic, cadmium, mercury, chromium, and thallium, due to their non-biodegradability and persistence in the environment. These share a high level of toxicity to aquatic organisms (e.g. copper) and carcinogenic or neurotoxic effects to humans (e.g. lead and mercury) even at low concentrations (Sood et al. 2012).

Mining is by far the biggest source of heavy metal contaminants of the environment for many countries involved in such activities and particularly in developing countries (Kalin et al. 2006). The issue of acid mine drainage (AMD) is at the centre of ecological problems associated with mining and it affects about 70 % of the world's mining sites (Global Capital Magazine 2008). AMD is formed when metal sulphides (e.g. pyrites) from mining solid wastes rocks are exposed to water and oxygen which results in dissolved metals and H_2SO_4 that cause a low pH (<4) in the tailings environment. Consequently this leads to leaching and increased metal mobility from mine tailings (Dudka and Adriano 1997). The AMD crisis in the province of Gauteng (South Africa) has been one such issue of environmental pollution over the last decade with AMD flooding the western basins of the Witwatersrand (McCarthy 2010). In Papua New Guinea, contaminated mine wastes from the Ok Tedi, Porgera and Tolukuma mines are discharged directly into local rivers (Christmann and Stolojan 2001). The upper North Branch of the Potomac River between the border of western Maryland and West Virginia in the USA is reported to have a poor water quality as a result of acid mine drainage from abandoned coal mines (Sheer et al. 1982). Acid mine drainage directly contaminates a total of 700 km of streams and rivers and more than 2300 ha of lakes and reservoirs in the USA (Cohen 2006). Similarly, the rapidly declining surface and ground water quality as a result of effluents and decants from abandoned mines in Gauteng and the North West Provinces (South Africa) has raised alarms over the last five years (van Eeden et al. 2009). The gold mines in the West Rand and Far West rand near Johannesburg, alone discharge an estimated 50 tonnes of uranium annually into the receiving surface water courses (Coetzee et al. 2006).

While water scarcity problems continue to rise as a factor of increasing world population and unpredicted impacts of global climate change, industrial and mining waste effluents remain the main concern of governments and environmentalists in water related issues. Thus, while increasing our water use efficiency and reducing our domestic and economic footprints on water resources is of immense importance and a matter of urgency, implementing an effective remediation technology in the abatement of water pollutants could make an important contribution to water security.

Conventional methods of remediation

A wide range of traditional methods for treating industrial and mine wastewaters are used to remove both organic and inorganic contaminants before their discharge into receiving watercourses. Ion exchange, reverse osmosis and electro-dialysis are used to remove nitrates from contaminated waters (Shrimali and Singh 2001). The same methods are also used in removal of heavy metals from water in addition to other methods such as chemical precipitation, coagulation-flocculation, floatation, ultrafiltration, activated carbon adsorption, and solvent extraction (Kurniawan et al. 2006). The effectiveness of each of these remediation methods however, depends on a number of factors, among which are the type and the concentration of the pollutants in the target solution. Heavy metals such as zinc, cadmium and manganese can be completely removed by chemical precipitation using lime treatments (Charerntanyarak 1999). The same method however, does not achieve complete removal of lead, or mercury contaminant from water unless pre- or follow-up treatments (e.g. reducing the solution with soda ash or sodium sulphide) are implemented (Dean et al. 1972).

Like many other techniques the traditional remediation methods have some limitations. Complete removal of contaminants is not achievable by most methods (Dean et al. 1972) and the massive amount of sludge and other residues generated in the process of mine effluent treatment raises the issue of their safe disposal into the environment (Rebhun and Galil 1990). In Canada, such sludge wastes reach an estimated 6.7 million cubic metres annually, which is simply the transformation of one form of waste into another, then released into the environment (Hall 2012). The pressing issue with traditional methods of remediation is however, the cost associated with them, which often discourages the moral and financial liability of mining companies to address their environmental footprints. While there is an urgent need for the development of new techniques to effectively reduce water pollution of all kinds, the use of green plants to clean contaminated water has been widely publicized and accepted as a potential solution.

Phytoremediation

Phytoremediation is the reduction of harmful contaminants in the environment to safer concentrations using green plants (Pivetz 2001; Garbisu and Alkorta 2001; Gratäo et al. 2005; Sharma et al. 2014; Emmanuel et al. 2014). Although, the inception of this concept goes back over three centuries, it has been revived as a new innovative technology for environmental rehabilitation and has had greater public acceptance from the mid 1970s onwards (Henry 2000). This is largely attributed to the fact that phytoremediation is a green and cost-effective technology compared to the conventional methods of remediation (Rahman et al. 2007; Suresh and Ravishankar 2004; Sood et al. 2012; Emmanuel et al. 2014; Rai 2009; Gratäo et al. 2005; Sharma et al. 2014). The USA is leading the world in phytoremediation with the potential value of the market estimated between US$33.8 and 49.7 billion annually, and similar companies are rising fast in Europe and Canada (Suresh and Ravishankar 2004). Although a true cost comparison between the conventional remediation and phytoremediation methods has not yet been well established for removal of water pollutants, there are few anecdotal examples in the literature. For example phytoremediation of contaminated soils costs 2–8 times less than the current conventional technology used. Similarly, the cost of phytoremediating contaminated water could be 7–50 times less than the traditional methods (Table 1).

Hyperaccumulators and accumulators

Some plants are naturally capable of accumulating heavy metals in their shoots, at concentrations between 100–000 times greater than normal non-accumulator plants, without any symptoms of stress (Manousaki et al. 2009; Kadukova et al. 2008). Reeves and Baker (2000) refer these 'absolute metalophytes' as hyperaccumulators and they have identified over 400 of such vascular plant species in at least 45 different families worldwide, of which Brassicaceae, Flacourtiaceae, Caryophylaceae, Cyperaceae, Cunouniaceae, Fabaceae, Lamiaceae, Poaceae, Violaceae, and Euphobiaceae are among these included in the list (Gratäo et al. 2005). The members of the Brassicaceae family constitute one of the most important groups of hyperaccumulators since they are capable of hyperaccumulating several metal elements in their shoots (Prasad and Freitas 2003). For instance, *Thlapsi caerulescens* (J. & C. Presl) is found to hyperaccumulate Cd, Co and other trace metals besides zinc, if the plant is exposed to these metals concurrently (Baker et al. 1994).

Phytoremediation is a broad term that encompasses several methods and among them are phytoextraction, rhizofiltration, phytovolotalization, phytostabilization, phytodegradation, and rhizodegradation (Vangronsveld et al. 2009). The most widely used method for removing and reducing heavy metals and metalloids from polluted soils is however, phytoextraction, which involves the removal of contaminants from the soil via the plant's roots and their accumulation in their harvestable biomass, followed by safe disposal (Salt et al. 1998). Ideally, hyperaccumulators would fit this method. Nevertheless, there are only a handful known of such species in the world, many of which are geographically restricted. Thus, many other non-hyperaccumulator plants, with fast growth and a large plant biomass can trade-off against their relatively low metal accumulation capabilities and have been selected as candidates for phytoremediation. Corn *Zea mays* (L.), sorghum, *Sorghum bicolour* (L.) Moench, alfalfa, *Medicago sativa* L., and willow trees (*Salix* spp.) are a few such examples (Pivetz 2001). As far as phytoremediation of polluted waters is concerned, however, the only applicable method is rhizofiltration, a sub-category of phytoremediation,

Table 1 Comparisons between the cost of phytoremediation and traditional (physical and chemical remediation) methods of remediation

Contaminant	Phytoremediation cost (US$/unit area)	'Traditional' remediation cost (US$/unit area)	Depth of soil (cm)	Source
Pb	6/m^2	15/m^2 –730/m^2	60-cm deep soil	Berti and Cunningham 1997
Cd, Zn, Cs	60,000–100,000/acre	>400,000/acre	50.8-cm deep soil	Salt et al. 1995
Unspecified contaminant	250,000/acre	660,000/acre	610-cm deep aquifer	Gatliff 1994
Petroleum	2500–15000/ha	20,000–60,000/ha	15-cm deep soil	Cunningham et al. 1996
Unspecified contaminant	0.02–40/kilolitre	1–300/kilolitre	Water	Weiersbye 2007

where contaminants are removed by absorption, adsorption or precipitation and are accumulated in or on the plant roots (Tomé et al. 2008). It is the method best-suited for cleanup of contaminated waters and is carried out by aquatic macrophytes, since the remaining of the phytoremediation methods are associated with terrestrial plants only (Pivetz 2001).

Aquatic macrophytes in phytoremediation

According to their growth forms in relation to the growth substratum, aquatic plants are categorized into four major groups (Brix and Schierup 1989; Rai 2009; Sood et al. 2012):

- Emergent macrophytes: with roots embedded in the soil and shoots growing above water, e.g. *Phragmites australis* (Cav.) Train, ex Steud., *Typha latifolia* L. (TL).
- Floating leaved macrophytes: growing on sediments submerged at a depth range of 0.5–3.0 m, e.g. angiosperms such as *Potamogeton pectinatus* (L)., water lilies *Nuphar* and *Nymphea.*
- Submerged macrophytes: occur entirely below the water surface, e.g. the obligate aquatic green algae, the charophytes, a few vascular plants the pteridophytes such as *Ceratophyllum demersum* L. (coontail), and many flowering plants, the angiosperms such as *Vallisneria spirallis* (L.), and *Hydrilla verticillata* (LF).
- Free-floating macrophytes: the roots float freely and no roots are anchored in the substratum, e.g. *Eichhornia crassipes*, *Salvinia* sp., *Azolla* sp., and *Lemna* sp.

Some of the main limitations of phytoremediation are: the extent of the plant's root system in relation to the depth of the contaminant occurrence; the growth period required to reach a well differentiated system of roots and shoots, the kind and concentration of heavy metal contaminants and tolerance of plants to metal toxicity (Pivetz 2001). Nevertheless, aquatic plants are relatively easy to propagate and grow fast, accumulating a large biomass within a short period. In fact most of the aquatic plants researched for their phytoremediation ability are often invasive and resilient to nutrient deficiency and environmental variability. Among these are *Eichhornia crassipes*, *Azolla* sp, *Lemna* spp, and *Myriophyllum aquaticum* (Vell) Verdc., *Ceratophyllum demersum*, *Hydrilla verticillata* (L.F.) Royle, *Phragmites australis*, *Typha latifolia*, *Arundo donax* (L.), *Vallisneria spiralis* (L.). They have an extensive root system and root surface area for uptake and removal of water contaminants, which occurs by adsorption of cations onto the negatively charged root surfaces (Elifantz and Tel-or 2002; Kivaisi 2001). The fact that most of the heavy metals removed by aquatic plants are accumulated in their root systems means that the plant's susceptible photosynthetic tissues are out of reach of metal toxicity, unlike in many terrestrial plants. Thus, aquatic macrophytes are more tolerant, effective and suitable for phytoremediation of water contaminants and particularly for treatment of domestic effluents and wastewaters than terrestrial plants (Sood et al. 2012). It is no surprise therefore to see an explosion in researches and reviews of aquatic plants as potential tools of phytoremediation in the last two decades, particularly in the first six major aquatic macrophytes mentioned above. This could also be due to their widespread occurrence across the major fresh water bodies of the world, often as invasive weeds, and their persistence despite the massive efforts directed at their control (Rai 2009).

Although much research on the ability of aquatic macrophytes to clean metal contaminated waters is commonly conducted in a controlled environment at a laboratory scale, they have all shown a high level of efficiency and relatively greater capacity for metal accumulation in their tissues compared to terrestrial plants. This is because metal contaminants are more bioavailable in water than in the soil where aquatic macrophytes have direct access to them (Sood et al. 2012).

Rai (2008b) investigated the metal removal efficiency of the free-floating macrophyte, *Azolla pinnata* (R.) Br., in an aquarium with varying concentrations of 0.5, 1 and 3 mg l^{-1} of Hg and Cd in isolation, and found 90, 94 and 80 % removal for Hg and 90, 91 and 70 % removal for Cd, respectively after 13 days of exposure. Other similar laboratory studies also found 93 % removal of Hg by *Azolla caroliniana* (Willd.) after 12 days (Bennicelli et al. 2004).

Among the other aquatic macrophytes researched extensively for phytoremediation are duckweeds, *Lemna* spp. They are among the few free-floating aquatic macrophytes that have been used in constructed wetlands for removal of heavy metals (Vaillant et al. 2004; Wang et al. 2002; Zayed et al. 1998). The two common species of the duck weed often cited in the literature are: *Lemna gibba L*, and *Lemna minor*, L. Mkandawire et al. (2004), found removal of 84.5 % of uranium and 88.2 % of arsenic by *Lemna* from contaminated water after 21 days of exposure. *Lemna* spp. has been occasionally indicated as a hyperaccumulator of heavy metals (Kara et al. 2003; Vaillant et al. 2004; Mokhtar et al. 2011) because of their ability to accumulate enormous amount of such contaminants in their tissues. *Lemna gibba* was found to grow naturally on tailing ponds of abandoned uranium mines, with 186.0 ± 81.2 μg/l uranium and 47.37 ± 21.3 μg/l arsenic concentrations greater than the background reference sites with 7.9 μg/l and 3.02 μg/l, respectively (Mkandawire et al. 2004). Many other aquatic plants had also been investigated for their potential as a tool of phytoremediation. (See Dhir et al. 2009).

The bioconcentration of metals by different aquatic macrophytes is variable but usually exceeds the concentration of metals in the occupied water by >100,000 times (Cardwell et al. 2002). Kumari and Tripathi (2015) investigated the emergent macrophytes, *P. australis* and *T. latifolia* in glass aquarium (75 L) with known concentrations of Cu, Cd, Cr, Ni, Fe, Pb and Zn metal contaminants collected from five different "sampling stations of untreated urban sewage mixed with industrial effluents" along the river Ganga at Varanasi, India and found an average of 40 to 57 % removal of the contaminants at the end of the experiment in day 14. Other halophytic plants such as *Sarcocornia fruticosa* (L) A.J. Scott., *Halimione portulacoides* (L.) Aellen, and *Spartina maritima* (Curtis) Fernald, also accumulate 9 fold of concentrations Hg and 44 fold MeHg (methylmercury) in their roots from coastal wetlands (Canario et al. 2007). However, the relationship between the amount of metal uptake by emergent wetland macrophytes and the metal concentrations in the underlying sediments is generally poor and inconsistent (Dunbabin and Bowmer 1992; Keller et al. 1998; Cardwell et al. 2002). Nevertheless, some emergent macrophytes show a predictable affinity for selected metal contaminants. The amount of Cu, Ni, Fe and Mn sequestered in the roots of the emergent aquatic macrophyte, cattails (*Typha latifolia* (L.)) were directly correlated with their concentrations in the sediment where they grew (Taylor and Crowder 1983). Deng et al. (2004) also found similar correlation with the uptake of Pb, Zn and Cu by the emergents, *Leersia hexandra (*Swartz.), *Equisetum ramosisti* (Desf.) and *Juncus effuses* (L.) from mine effluents in China.

Some submerged macrophytes also show a positive correlation between the bioconcentration of metal contaminants and their sediment concentrations. Chen et al. (2015) found an increase in the accumulation of heavy metals in the tissues of the submerged rootless macrophytes, *C. demersum* with the increase in Pb concentrations when the plants were exposed to five different Pb solutions (5–80 μM). They found a maximum accumulation of 4016.4 mg/kg dry weight (dwt) of plant biomass. Similarly, the accumulation of Ni in the tissue of the submerged plant, *H. verticillata* (LF) Royle increased from a concentration of 40 μg/g when exposed to a Ni solution of 5 μM to 502, 1198, 1474, 2168 and 4684 μg/g dwt at solutions of 10, 25, 50 and 100 μM of Ni, respectively after six days (Sinha and Pandey 2003). The bioconcentration of metals is also relatively higher in submerged macrophytes than the emergents or other aquatic macrophyte groups (Albers and Camardese 1993). Dogan et al. (2015) compared two submerged macrophytes (*C. demersum* and *Rotala rotundifolia* (Roxb.) Koehne) and the emergent *Bacopa monnieri* (L) Pernnell, in the removal of Cd from an aqueous solution with concentrations of 0.1, 1 and 10 mg/l and found the first two submerged macrophytes accumulated more cadmium than the emergent macrophytes, *B. Monnieri* with concentrations of 825, 1459 and 757 mg/g dry weight, respectively.

The macroalgae in the family of Characeae are also among the aquatic macrophytes, with a potential for wastewater treatment. They have high tolerance to heavy metals, and grow through autotrophic and heterotrophic modes of nutrition, and have a large surface area, through which they detoxify heavy metals by complexing them into phytochelatins (González et al. 2007). Most species in the family are found in two genera, *Chara* and *Nitella* (Meurer and Bueno 2012). Al-Homaidan et al. (2011) found a concentrations of 339 Mn, 44 Cu and 69 As μg/g dwt in the thali (plant body) of *Enteromorpha intestinalis* (Linnaeus) Nees and 211 Mn, 66 Cu and 8 As μg/g dwt in *Cladophora glomerata* (Linnaeus). The macroalgae *Chlorophyta* is known as a hyperaccumultor of As and Boron (B) (Baker 1981).

Among the aquatic macrophytes, the only group with a limited research for phytoremediation is the floating-leaved macrophytes. Nevertheless, some studies have already shown their potential for removal of metal contaminants and their use for phytoremediation. For instance, Choo et al. (2006) tested the removal of chromium, Cr (VI) from aqueous solutions with five different concentrations ranging from 1–10 mg/l using the tropical water lily, *Nymphaea spontanea*. They found a removal of >60 % of Cr within seven days and metal accumulation in the plant's tissues increased with the increase of the Cr concentrations in the solution.

Water hyacinth, *Eichhornia crassipes* (Mart.) Solms-Laubach (Pontederiaceae) is native to the Amazonian region in South America (Harley 1990). It is the world's worst aquatic weed. Water hyacinth is resilient to a wide range of climatic conditions and can survive temperatures between 1–40 °C and extremes of water nutrient levels (Malik 2007). Water hyacinth is also prevalent in waters contaminated with trace amounts of heavy metals and other inorganic and organic contaminants from mining and industrial wastewater discharges. The water quality of the 750 km long Lerma River in west-central Mexico is highly compromised by wastewaters discharges from 20 urban municipalities and over 2500 industries in the course of the river, making it one of the most polluted waters in the country (Tejeda et al. 2010; Helmer and Hespanhol 1997; de México 2000). Nevertheless, water hyacinth is one of the few aquatic plants prevalent in this river (Tejeda et al. 2010).

The wide geographical spread of water hyacinth and its ability to have a high biomass turnover within a single growing season, coupled with its resistance to elevated concentrations of organic and inorganic water contaminants, makes it one of the most widely tested plants for phytoremediation, particularly among the aquatic plants (Brooks and Robinson 1998; Vymazal 2008). The effectiveness of water hyacinth in the removal of both organic and inorganic water contaminants has been tested on a number of occasions and usually a reduction of over 80 % in contaminants had been reported (Table 2).

Aquatic macrophytes in constructed wetlands

The mobility of metal contaminants in soil depends on several factors, among which are, concentrations, chemical form, metal property, binding state, organic matter, pH and root exudates. For instance, arsenic in mine affected soils binds with

Table 2 The phytoremediation capacity of water hyacinth (Adapted and modified from Newete, 2014)

Wastewater source	Metal removed from water	Removal from water (%)	Duration of experiment (days)	Reference
Coal mine effluent	As	80.00	21	Mishra et al. 2008a
Contaminated solution (1.5 mg Cu/L)	Cu	97.00	21	Mokhtar et al. 2011
Textile effluents	Cr	94.78	4	Mahmood et al. 2005
Textile effluents	Zn	96.88	4	Mahmood et al. 2005
Coal mining effluent	Cd	66.4	21	Mishra et al. 2008b
Coal mining effluent	Fe	70.5	21	Mishra et al. 2008b
Contaminated solution (1 mg Hg/L)	Hg	99.9	30	Newete 2014
Contaminated solution (1 mg Mn/L)	Mn	78.4	21	Newete 2014
Contaminated solution (1 mg U/L)				Newete 2014
Contaminated solution (0.8 mg NO_3^-N/L)	NO_3^-N	62.00	1	Petrucio and Esteves 2000
Contaminated solution (0.6 mg NO_3^-N/L)	PO_4^-P	68.20	1	Petrucio and Esteves 2000

Fe and Mn oxides or is retained as sulphides (Moreno-Jiménez et al. 2010; 2011). Over 70–90 % of arsenic is found in its inert form in soils contaminated by mines (Conesa et al. 2008). Thus, unlike soil contaminants, water contaminants are relatively bioavailable and readily accessible for phytoremediation. As a result aquatic plants are more effective for phytoremediation than terrestrial plants (Brooks and Robinson 1998) and have widely been implemented.

There are at least 650 constructed and natural wetlands in North America and over 5000 of them in Europe (Kivaisi 2001). The dominant forms of aquatic plants in most wetlands are the emergent aquatic macrophytes (Vymazal et al. 1998) which are suitable for temperate regions (Nahlik and Mitsch 2006) because free-floating aquatic plants such as water hyacinth is affected by frost in cold temperate regions (Vymazal et al. 1998).

Although constructed wetlands were primarily designed to improve the water quality of domestic, municipal and agricultural wastewaters, they have evolved over the years and been extended to include industrial and mine wastewater treatments. Natural and constructed wetlands with emergent aquatic macrophytes such as reeds (*Phragmites australis*), cattails (*Typha* spp.), and bulrushes (*Scirpus* spp. and *Schoenoplectus* spp.) have been used effectively in the treatment of domestic effluents, mine and industrial wastewaters with heavy metals contaminants (Yang et al. 2006). A wetland constructed with *Typha latifolia*, *Phragmites australis* and *Cyperus malaccensis* (Lam.) in 1983 for the treatment of Pb/Zn mine discharges, at Shaoguan in Guangdong Province (China) successfully 'polished' the wastewater and significantly improved the water quality by removing 94 % of Cd, 99.04 % of lead (Pb), 97.30 % of zinc (Zn), and 98.95 % of total suspended solids (TSS) from their initial concentrations of 0.05 mg/l Cd, 11.5 mg/l Pb and 14.5 mg/l Zn, all of which were well above the legal industrial wastewater limits (Yang et al. 2006). Similarly a constructed wetland with cattails, *Typha Latifolia* L., at Springdale, Pennsylvania (USA) is used for the treatment of iron (Fe), manganese (Mn), cobalt (Co) and nickel (Ni) contaminants from an electrical power station and has achieved a reduction of up to 94, 98, 98 and 63 %, respectively over two years (Ye et al. 2001). The success of the method is such that the USA has 400 constructed wetlands exclusively for the treatment of coal mine waste water drainage (Perry and Kleinmann 1991).

The underlying sediments of wetlands are the largest sink of most metal contaminants (Ye et al. 2001). This suggests the use of rooted submerged macrophytes, besides the emergents, is more suitable candidate for phytoremediation than the free-floating aquatic macrophytes that only absorb/adsorb metals from the water column. The submerged macrophytes are however, considered to be more efficient in metal accumulation than the emergent macrophytes (Albers and Camardese 1993) because of the large surface area of the entire plant biomass in direct contact with contaminants in the water system (Xing et al. 2013). Nevertheless, the practical function of the submerged macrophytes and floating leaved macrophytes are still in its infancy stage and is not yet implemented or developed (Bashyal 2010).

Weeds for phytoremediation

Although water hyacinth and duckweed are the two plants predominantly used in constructed wetlands, particularly in tropical and subtropical regions (Kadlec and Knight 1996; Bashyal 2010), their invasive nature makes their application as a phytoremediation tool controversial and subsequently they have not yet been fully exploited properly despite the intensive research conducted on their potential as tool of phytoremediation. Nahlik and Mitsch (2006) compared seven species of aquatic plants including the dominant free-floating macrophytes water hyacinth and water lettuce (*Pistia stratiotes* L.), in various constructed wetlands for the treatment of wastewaters from a dairy farm, a dairy processing plant, a banana paper plant, and a landfill in the Parismina River Basin in eastern Costa Rica. The concentration of ammonium in the constructed wetlands was reduced by 92 % and Phosphorus by 45–92 %. Similarly, Maine et al. (2007) used water hyacinth in a large constructed surface wetland for the treatment of wastewaters with Cr, Ni and Zn contaminants from a tool factory in Santo Tomé, Santa Fe, Argentina which effectively removed 89, 93 and 99 % of the contaminants respectively, although in the second year *Typha domingensis* (Pers.) was incorporated into the wetland to replace the declining population of the water hyacinth as a result of elevated metal toxicity.

Compared to water hyacinth, the inclusion of the duckweed species in constructed wetland is more limited due to their reduced roots, for direct exposure to the contaminants and small root surface area for the attachment of microorganisms involved in the remediation process (Kivaisi 2001). Thus, they are often limited to small scale surface water structures and lagoons (Vymazal et al. 1998; Bashyal 2010).

While a selection of appropriate plants, based on their tolerance, rate of biomass turnover, and their efficiency in the abatement of wastewater is of a paramount importance, the safe disposal of the phytoremediating plants is an issue that has to be addressed and this will be reviewed for aquatic macrophytes with particular reference to water hyacinth.

The fate of water hyacinth after phytoremediation

Phytoremedaiton has been labelled by many researchers as an emerging, cost-effective and environmentally friendly method for the rehabilitation of polluted environments (Sharma et al. 2014; Rai 2008a; Garbisu and Alkorta 2001; Sood et al. 2012; Emmanuel et al. 2014; Rahman et al. 2007). While this is true in many aspects compared to conventional methods of remediation, it has its own drawbacks. Fast growth and biomass production is good for the efficacy, but plant seasonality (Rai 2008a; Maine et al. 2007) and poor tolerance to high metal concentrations is a constraint on the technology (Mannino et al. 2008). Thus, unlike domestic wastewater treatment, aquatic plants in constructed wetlands are used in secondary or tertiary industrial and mine wastewater treatments, because of the high concentration of heavy metals and their toxicity to the plants (Avsara et al. 2007; Sharma et al. 2014; Susarla et al. 2002). Furthermore, effective phytoremediation processes should involve a regular harvest and safe disposal of plants (Rai 2008a), particularly with aquatic macrophytes, since they will eventually die, decompose and then release the elements sequestered, back to the source more rapidly than terrestrial plants would (Rai 2008a). However, despite increasing research in the field of phytoremediation, the issue of safe disposal of phytoremediating plants has rarely been addressed.

The harvest and disposal of plants (usually weeds) removed from heavily infested aquatic waters, whether such plants have been used for the purpose of phytoremediation or not, is often expensive and discouraging. As a result several attempts have been made to convert the harvested waste plant biomass into economically beneficial material to offset the cost of harvest and disposal. One common example is biogasification of harvested waste plant biomass. The use of some aquatic macrophytes such as water hyacinth as biofuel is well established (Rahman and Hasegaw 2011; Isarankura-Na-Ayudhya et al. 2007; Awasthi et al. 2013; Bergier et al. 2012; Bhattacharya and Kumar 2010; Gunnarsson and Petersen 2007) often in an attempt to deal with aquatic plant biomass after their removal from invaded water systems. However, the process of economically viable production of ethanol from water hyacinth biomass is complicated by the presence of a considerable amount of hemicelluloses, cellulose, and lignin components (Abraham and Kurup 1997), which constitutes 35, 25 and 10 % of the plant dry matter, respectively (Gunnarsson and Petersen 2007). Thus, to optimize the extraction of fermentable soluble sugars, the plant biomass has to undergo pre-treatment prior to the actual process of scarification, and microbial fermentation to produce ethanol (Abraham and Kurup 1997; Cheng et al. 2014; Masami et al. 2008; Bhattacharya and Kumar 2010). However, while directing the biomass waste of aquatic macrophytes into a source of biofuel is highly publicized, it is still in its experimental stage and its economic viability is confounded by the cost of pre-treatment reagents and the lack of a single prescription for such reagents in processing the biomass of different aquatic macrophytes (Mishima et al. 2006; Awasthi et al. 2013). Furthermore, the potential technology of generating biofuel from such plants does not address the issue of disposal of the heavy metals in the plant tissue of the aquatic macrophytes have been used in phytoremediation of industrial and mine wastewaters.

Other disposal methods include briquetting or carbonization to make charcoal, and incineration, (Rahman and Hasegawa 2011). Although, water hyacinth can be sun dried for incineration to use directly as source of energy (e.g. cooking fires), its commercialization beyond a small scale production is curtailed by the fact that 90 % of the plant biomass is made of water (Abdelhamid and Gabr 1991) and the amount of energy produced is less than 1.3 GJ/m^3 compared to the same volume of charcoal (9.8 GJ/m^3) (Gunnarsson and Petersen 2007). Improving this method by compacting the dried water hyacinth into briquettes or pellets produces about the same amount of energy (8.3 GJ/m^3) that the same volume of charcoal can produce (9.8 GJ/m^3) (Thomas and Eden 1990). This could work, but the initial investment in machinery, and the cost of large areas required for drying plant biomass followed by their transportation to the site of production is not encouraging and requires proper evaluation (Rahman and Hasegawa 2011). The obvious limitation of the method is however, the large amount of ash produced (40 % for water hyacinth) (Rahman and Hasegawa 2011) compared to the average ash content of between 0.5 and 5 %, depending on the wood species and materials (commonly Sawdust, planer shavings and dry chips) used to make pellets (Lehtikangas 2001). In addition, using briquettes made from water hyacinth contaminated by heavy metals for domestic purposes could lead to health hazards. For instance incineration of arsenic contaminated water hyacinth could be a source of air pollution and related health problem (Rahman and Hasegawa 2011).

Water hyacinth biomass as a compost

The use of water hyacinth as a compost to improve soil structure and nutrient could be an option in the management of waste biomass and particularly in developing countries, where the artificial fertilizers are often not affordable (Gunnarsson and Petersen 2007). Water hyacinth retains a considerable amount of nutrients such as N, P and K and making water hyacinth compost takes a relatively short period (less than 30 days) (Polprasert et al. 1980) which makes it feasible for farmers seeking to improve their soil conditions. In the past water hyacinth compost was even commercialized by a company in Florida, USA which produced a finished compost from a mixture of equal proportion of water hyacinth and peat at a cost of $1.31 and sold for $1.75 per bushel in 1973 (Mara 1974). This could be a viable option for waste plant biomass treatment if the management target is only to address the infestation of water hyacinth. However, exposing phytoremediating aquatic macrophytes to a suite of heavy metals, which are then used as a compost to improve soil nutrients, would simply mean relocating the environmental problem from point A to point B.

Disposal of phytoremediating plants in mine tailings dams

Mine tailings dams The impoundment of mining waste into tailing storage facilities (tailings dams) and the associated problems of acid mine drainage in surface and ground waters are of a major concern as a result of runoff, infiltration, and leaching, or even a collapse of the tailing dams either due to poor design or earthquake. For instance the Ok Tedi gold and copper mine dam failure in 1984 in Papua New Guinea led into a devastating environmental impact with annual discharges of 60 million tonnes of tailings into the Fly River and the Gulf of Papua for many years (Cooke and Johnson 2002). Such incidents of major tailing dam failure are reported to occur between 2 to 5 a year at least for the last three decades (Davies 2002). According to Davies and Martin (2000) the total number of tailing dams in the world is estimated to be over 3500. Of this approximately 400 of them are found in South Africa (van Wyk 2002) which were erected since the start of gold mining on the Witwatersrand in 1886, and which collectively have accumulated an estimated 6 billion tonnes of tailings (Winde and van der Walt 2004). The USA generates an estimated 2 billion tonnes of solid wastes from mining operations annually (White 2003). Similarly mine waste in tailing dams was estimated to be 265.4 million tonnes in 2002 in China (Li 2006). The disposal of waste from the mining of silver, cadmium, copper, indium, sulfuric acid and zinc since 1966 in tailing dams at Kidd Creek mining in northern Ontario, Canada is expected to reach over 130 million tonnes by 2023 when mining at the site closes (Hudson-Edwards et al. 2011).

Although many countries have adopted stricter laws and measures that force mining companies to reduce their environmental footprint, the rehabilitation of mine solid wastes in tailing dams after mine closure is very slow and is expected to last more than a 1000 years (Szymanski and Davies 2004; Chambers and Higman 2011). For instance the Goldenville mine at Nova Scotia, Canada that was once operational between 1860 and 1945 still has 3 million tonnes of mine solid waste in tailing dams left behind after the mine closure (Müezzinoğlu 2003). Mine tailings dams are therefore, here to stay at least for the foreseeable future and their number will keep rising, since there is more waste rock generated for the same amount of a precious metal than was the case in the last century. The average copper ore grade has dropped to 0.5 % in 1975 from an average of 4 % in 1900 (Cooke and Johnson 2002). Consequently such mining escalation has raised the amount of tailings generated globally from 17 to 290 Mt per annum within the same period (Williamson et al. 1982). Thus, intensive remedial efforts and effective restorations methods for mine tailings dams are essential to reduce their environmental impacts.

The best and internationally accepted restoration practice is the levelling of tailing dams followed by revegetation with native plants improving the soil's physical and chemical properties. However, the cost of such total restoration of tailing dams is expensive and could exceed the total income generated by the mine (van Wyk 2002). Consequently, the economically viable option and ecologically sound approach for many of these tailing dams remains the revegetation of the slopes with native plants (Mendez et al. 2007). The first such practice in gold mine tailings dams in South Africa started in 1894 (Gunn 1973). Nevertheless, a complete vegetation cover and successful establishment of functional ecological systems could not be achieved due to the hostile soil properties of the tailing dams for plant growth (Cook 1971).

Tailing dams lack one of the main structural components of the soil profile, the topsoil and soil organic matter (Wanenge 2012; Wong 2003). This together with soil properties such as low pH, high silt content, increased toxic metal concentration, high erosion and poor nutrient levels, make the environment of tailing dams inhospitable for plant growth (Mendez et al. 2007; Cooke and Johnson 2002; Witkowski and Weiersbye 1998). As a result, improving the soil's physical and chemical properties and soil microbial activities is an integral process that precedes the revegetation processes. Phytoremediating plants disposed on mine tailings dams could therefore be used as mulches for soil amendments, while the metal contaminants in the plant tissues are released back to the source of water contamination, the mine tailings dams.

Mulching and decomposition on mine tailings dams The elevated heights of tailings dams above the natural ground surface expose the soil of the storage facilities to wind dispersion and water erosion (Witkowski and Weiersbye 1998). Thus, short term revegetation of tailings dams was initially conceived to restrict wind and runoff erosion and to minimize environmental pollution. Such revegetation however, was eventually adopted as the long term solution to the growing number of associated environmental hazards (Johnson et al. 1994; Carroll et al. 2000). Sewage sludge, organic compost and mulches are among few soil additives applied to improve the soil properties of mine tailings dams to enhance plant growth (Okalebo et al. 2006). Sewage sludge has more nutrients and can improve tailing dams' soil fertility faster than organic compost and mulches. However, due to their high heavy metal content, compost and mulches are the preferred soil amendment materials used for agricultural and mine tailings dam soils (Wanenge 2012).

Water hyacinth is a notorious invasive alien plant outside its native geographical locations and a widely used plant for phytoremediation. Its fast growth makes the plant an extraordinary sink for nutrients and an important mulch and soil fertility improvement in low nutrient soils. According to Reddy and D'Angelo (1990) the carrying capacity of water hyacinth, which is the maximum biomass of a species supported per unit area (Maler 2000), is 70 kg/m^2, although the time taken to reach such carrying capacity largely depends on environmental factors such as temperature and nutrient levels. Hauptfleisch (2015) compared the time taken to reach the carrying capacity of water hyacinth at two sites in South Africa, namely Delta Park and Mbozambo Swamp. While the first with a maximum growth rate of 0.053/g/g/day and a minimum growth rate of −0.004/g/g/day took 315 days to reach the carrying capacity, the latter took only 92 days at growth rates of 0.058/g/g/day and 0.024 g/g/day, respectively. The difference is attributed to the fact that Mbozambo Swamp is warmer and more eutrophied than the Delta Park (Byrne et al. 2010). Similarly, Amoding et al. (1999) found the water hyacinth doubling time in Ugandan waters was between 4–7 days with the highest growth rate of 228 tonnes per hectare per year. They also found the nutrient content of the 33 ha plant biomass behind the dam at Owen Falls in Uganda was estimated to be 23.2 tonnes of N, 3.5 tonnes of P and 52.0 tonnes of K. Water hyacinth invasion is an environmental menace, but could be redirected for soil amendment to enhance the revegetation of mine tailings dams in remedial efforts.

While water hyacinth as mulch might protect soil from wind and water erosion and increase its water retention capacity, such advantages are often short lived due to its rapid decomposition (Brady 1990), even though such decomposition leads to rapid release of nutrients to the soil. For instance, the application of wet water hyacinth as mulches at a rate of 150 kg/ha to 450 kg/ha in a maize field in Rwanda led to an increased soil fertility and maize production compared to the control treatments (without mulch) (Gashamura 2009). Unlike agricultural soils, in mine tailings dams, rapid decomposition of mulches may not be a problem as a result of few soil microorganisms (Tomlin 2012). Litter decomposition depends on several factors among which are litter chemical composition, temperature, soil moisture, and the soil fauna which includes the soil organisms such bacteria, fungi and protozoa and nematodes and arthropods (Singh and Gupta 1977). The hostile soil environment of mine tailings dams predominantly characterized by low pH (<4) and toxic heavy metals however, inhibits, soil organisms which are consequently found in low numbers. Grigg (2002) found microbial biomass to be 3–5 times less in a tailings dam at the Kidston Gold Mine in north Queensland, Australia, than in the surrounding unmined soils. Thus, although the decomposition rate of water hyacinth mulch in mine tailings dams could be slow, improvement of soil fertility could be expected over a period of time. For instance, Wanenge (2012) tested five different tailings amendments among which were fresh and dry water hyacinth biomass applied as mulches in order to determine its effects on soil fertility, seed emergence and plant survival of different native plant species. He showed that most of the plant species tested generally performed well compared to those on the control tailings (which were not amended), where no plants grew at all. He also found tailings amended with 0.5 % fresh water hyacinth mulch induced the most favourable plant conditions compared to other amendments, such as sewage sludge, or dry water hyacinth. Similarly, Grigg (2002) found an overall litter weight loss of 52–63 % from both mined and unmined sites in a litter decomposition experiment after 80 weeks, although, the litter weight loss was greater in the latter. The build-up of microbial biomass generally takes more than 15 years on agricultural lands (Insam and Domsch 1988) and even longer in mine tailings dams. Nevertheless, the presence of a carbon source such as fresh organic matter or plant materials on tailings dams can facilitate the rapid build-up of soil microbial population faster. Thus, mulching of mine tailings dams with plant materials has a pivotal role in the improvement of soil in the mine tailings dam besides protection from soil erosion.

Cost of harvest and transportation of aquatic plants

Revegetation and ecological restoration of mine tailings dams has long been adopted as a viable option of remediation. The cost of harvesting and transporting phytoremediating aquatic plants from water to mine tailings dams for mulching is relatively cheap. Trouzeau (1972) estimated the transportation cost of water hyacinth in Florida, USA as \$0.27/tonne/mile. Similarly, the cost of water hyacinth removal from point A to point B determined from a 20 years of cost analysis, was estimated to be \$400/acre or \$2/tonne (Thayer and Ramey 1986). The harvesting cost of water hyacinth was estimated over the same period in Florida, from 23 mechanical harvesting contracts, to be about \$4 649/acre (Haller 1995). In another example the cost of mechanical and manual removal of water hyacinth including the running cost was estimated from a survey in the River Nile in Egypt as \$7 million annually (Labrada 1995). Considering the cost of conventional cleanup of mine tailings dams, the harvesting and transportation of water hyacinth after phytoremediation still remains economically feasible. For instance the cleanup of 55,7000 tonnes abandoned hard rock mines in the USA is estimated to cost the country between \$32 and 72 billion (Kleinman 1989) and \$2 to 5 billion dollars in Canada for the cleanup of 12000 hectares of tailings and 350 million tonnes of waste rocks accumulated in the past 50 years (Jennings et al. 2008).

Over 100 countries in the world are involved in mining of metals and minerals (excluding oil and gas) and the majority of this are in developing countries (Bond 2002) where the actual cleanup and restoration processes of mine tailings dams are less affordable. Thus, the disposal of aquatic macrophytes, including water hyacinth after their use in the abatement of mine and industrial wastewaters, to mine storage facilities such as tailings dams, which are largely the sources of most heavy metal contaminants of most surface and ground water bodies, could be a viable option so long as such sites exist. The Witwatersrand Basin in South Africa alone has over 270 such tailing dams stretching over an estimated 400 km^2, most of which are unlined and unvegetated and are sources of much environmental pollution in the region (Oelofse et al. 2007). Dumping of the plant biomass harvested after phytoremediation on such tailings dams could act as mulch to suppress dust dispersal from the dams, and their decomposition will release the heavy metals from the plants back to where they belong. In the process, the soil fertility will be reinstated and revegetation of the tailing dams is enhanced. This could however, be only a solution as long as such tailing dams are available for disposal.

Conclusion

Aquatic macrophytes have widely been used as a tool of phytoremediation of contaminated waters. Despite increased research on aquatic macrophytes for phytoremediation however, the safe disposal of the phytoremediating plants is not well established. Over half of the nations in the world are involved in mining of precious metals and other minerals. Tailings dams are the prominent waste storage facilities in such activities. The disposal of phytoremediating plants on slopes of these tailing dams could act as mulches to suppress dust, while decomposition would return the heavy metals back to where they belong and reinstate soil fertility for revegetation in the tailing dams.

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